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REVIEW ARTICLE

A survey of unresolved problems in life cycle assessment

Part 2: impact assessment and interpretation

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Abstract

Background, aims, and scope Life cycle assessment (LCA) stands as the pre-eminent tool for estimating environmental effects caused by products and processes from 'cradle to grave' or 'cradle to cradle.' It exists in multiple forms, claims a growing list of practitioners and remains a focus of continuing research. Despite its popularity and codification by organizations such as the International Organization for Standardization and the Society of Environmental Toxicology and Chemistry, life cycle assessment is a tool in need of improvement. Multiple authors have written about its individual problems, but a unified treatment of the subject is lacking. The following literature survey gathers and explains issues, problems and problematic decisions currently limiting LCA's impact assessment and interpretation phases.

Main features The review identifies 15 major problem areas and organizes them by the LCA phases in which each appears. This part of the review focuses on the latter eight problems. It is meant as a concise summary for practitioners interested in methodological limitations which might degrade the accuracy of their assessments. For new researchers, it provides an overview of pertinent problem areas toward which they might wish to direct their research efforts. Having identified and discussed LCA's major problems, closing sections highlight the most critical problems and briefly propose research agendas meant to improve them.

Results and discussion Multiple problems occur in each of LCA's four phases and reduce the accuracy of this tool. Considering problem severity and the adequacy of current solutions, six of the 15 discussed problems are of paramount importance. In LCA's latter two phases, spatial variation and local environmental uniqueness are critical problems requiring particular attention. Data availability and quality are identified as critical problems affecting all four phases.

Conclusions and recommendations Observing that significant efforts by multiple researchers have not resulted in a single, agreed upon approach for the first three critical problems, development of LCA archetypes for functional unit definition, boundary selection and allocation is proposed. Further development of spatially explicit, dynamic modeling is recommended to ameliorate the problems of spatial variation and local environmental uniqueness. Finally, this paper echoes calls for peer-reviewed, standardized LCA inventory and impact databases, and it suggests the development of model bases. Both of these efforts would help alleviate persistent problems with data availability and quality.

Keywords Environmental assessment · LCA · LCA methodology · LCIA · Life cycle impact assessment · Life cycle interpretation

Preamble This series of two papers reviews unresolved problems in life cycle assessment (LCA). Part 1 (Reap et al. 2008) focuses upon problems in the goal and scope definition and life cycle inventory analysis phases. Part 2 discusses problems in the life cycle impact assessment and interpretation phases. Having probed LCA's main weaknesses, Part 2 identifies critical problems and suggests research agendas meant to ameliorate them. Additionally, the second paper in the series brings closure to the review with a unifying summary.

Part 1 'Goal & Scope and Inventory Analysis' has been published in Int J Life Cycle Assess (2008) 13(4), DOI 10.1007/s11367-008-0008-x

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1 Background, aims, and scope

This work continues and concludes a literature review that gathers, lists, and concisely explains technical problems in life cycle assessment (LCA). Specifically, it discusses problems arising in LCA's impact assessment and interpretation phases. The International Organization for Standardization's (ISO) structure for LCA provides a familiar means of organizing the problems discussed in this survey. Problem descriptions are organized according to the LCA phase in which each arises. Beginning where Part 1 ended, Section 2 of this paper reviews problems occurring in the life cycle impact assessment phase. Section 3 documents and explores problems associated with the interpretation phase. Section 4 covers data quality and availability. In Section 5, the most pressing problems are identified and research programs meant to ameliorate them are proposed. Section 6 contains closing remarks.

2 Limitations of life cycle impact assessment

Translating burdens into environmental impacts is, arguably, the most challenging of LCA's four phases. The main problems faced during life cycle impact assessment (LCIA) result from the need to connect the right burdens with the right impacts at the correct time and place. Therefore, this section addresses the problems associated with impact category selection, spatial variation, local uniqueness, environmental dynamics, and decision time horizons.

2.1 Impact category selection

The mandatory elements for a LCIA involve the selection of impact categories, category indicators and models, the assignment of LCI results to the impact category (classification), and the calculation of category indicator results (characterization) (ISO 2000a). In this section, problems associated with all of these steps (except the characterization step) will be discussed in more detail as illustrated in Fig. 1.

2.1.1 Current difficulties with selecting impact categories

There are various practical difficulties currently associated with impact category selection. These difficulties spring from a lack of current standardization in several impact categories present in the LCA literature, despite efforts to standardize them (Udo de Haes et al. 2002). For instance, Finnveden noted the slightly different impact category lists that have been proposed by different organizations (Finnveden 2000). These differences could be due to the impact modeling approach taken (midpoint versus endpoint) or the categories selected. The lack of standardization of some impact categories is also demonstrated in the recent debate as to whether certain impact categories such as soil salinity, desiccation, and erosion should be their own category or part of another category such as land use impact and freshwater depletion (Jolliet et al. 2004). Similar discussions surround consideration of some impacts such as casualties due to accidents, with some suggesting the use of other tools in parallel (Jolliet et al. 2004).

Disuse is one consequence of a lack of standardization in some impact categories. For instance, Finnveden observed that impact categories such as land use, habitat alteration, impacts on biodiversity, nontoxicological human impacts, and impacts in work environment typically escape consideration (Finnveden 2000). This represents a potentially significant problem. As Hellweg and coauthors discovered, failing to consider chemical exposure at the workplace could lead to process optimizations at the expense of worker's health (Hellweg et al. 2005). Obviously, other reasons besides a lack of standardization may cause LCA practitioners to not consider certain impacts such as: (1) lack of data to support a proper assessment of that category, (2) the belief that the category is not relevant in the given study, and/or (3) lack of consideration of the category in the LCA methodology or tool used (e.g., if one used Eco-indicator 99). The first and last reasons are certainly sources of concern that need to be dealt with in order to improve the quality of LCA studies. Overall, these issues could create problems in the comparability of similar LCA studies directly influence the data gathering efforts and the quality of the results.

Another potential result affecting both the level of confidence or relevance for decision making of the LCA study results stems from whether the LCA practitioner selects midpoint or endpoint (damage) impact categories. For instance, endpoint categories are less comprehensive and have much higher levels of uncertainty than the better-defined midpoint categories (UNEP 2003). Midpoint

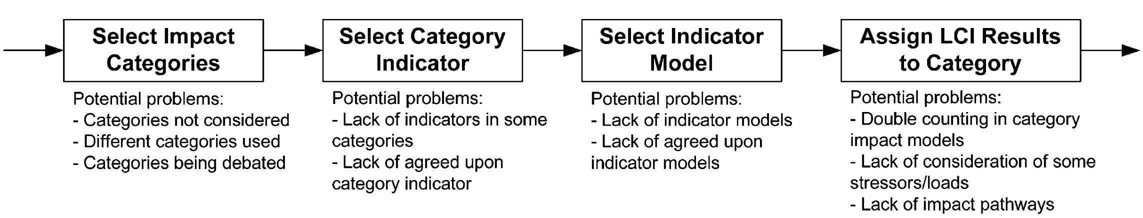

Fig. 1 Potential problems in impact category, indicator and model selection

categories, on the other hand, are harder to interpret because they do not deal directly with an endpoint associated with an area of protection as defined by Udo de Haes and coauthors (Udo de Haes et al. 2002) that may be more relevant for decision making, especially in policy (UNEP 2003).

2.1.2 Data gaps with toxicity-related impact categories

Typically, certain impact categories such as land use, habitat alteration, impacts on biodiversity, human toxicity and ecotoxicity, aquatic eutrophication, and photo-oxidant formation suffer from significant data gaps (Finnveden 2000). The development of more complete databases may fill some of these data gaps, but impact categories such as human toxicity and ecotoxicity are not expected to be greatly improved due to the large numbers of chemicals used by society and the potential synergistic effects between these chemicals (Finnveden 2000). This concern is strengthened with statements from other researchers that say uncertainties “are likely to remain high, despite consensus building” (Pennington 2001) and from researchers that compared the aquatic ecotoxicity impact results of three different LCIA methods for various detergents and found different answers from the methods (Pant et al. 2004).

2.1.3 Current difficulties with selecting impact category indicators and models

The previously noted problem of lack of standardization in the definition of impact categories is also seen when selecting an impact category indicator and model. In the case of some endpoint impact categories, such as damage to the abiotic natural environment (degrading of landscape, glaciers, waterfalls, etc.) and biotic natural resources (wild plants and animals used by humans), no indicator has been proposed (Jolliet et al. 2004). In other categories, such as damage to the biotic natural environment (wild plants and animals, ecosystems) and abiotic natural resources (destruction or dissipation of nonrenewable natural resources), damage indicators exist, but they have not been agreed upon (Jolliet et al. 2004). In the latter category, more than one method is available for abiotic deposits (Finnveden 2000), which leads to different models. This can be significant because resource use is an important category for popular LCA methodologies such as the EPS system and Eco-indicator 99 (Finnveden 2005).

2.1.4 Current difficulties with assigning LCI results to impact categories (classification)

Some environmental burdens impact multiple categories which introduces the potential for ‘double-counting’ (Hauschild and Wenzel 2000). Oxides of nitrogen (NO_x) can contribute to smog, acidification, and eutrophication, for instance. Continuing with the example of NO_x, double-counting occurs when one erroneously assigns the same NO_x emission to two or more categories instead of appropriately apportioning it. This type of error can occur as a result of a practitioner’s ignorance, but flaws in the models used to partition burdens among impacted compartments (Guinee and Heijungs 1993) are a more likely and less easily detected source of double-counting. Double-counting effectively magnifies the impact of a particular burden and skews a study's results, potentially leading to a poor decision during the interpretation phase. Therefore, practitioners should understand the appropriate chain of causality linking burden to impact and their models must accurately partition burdens.

In some cases, the effect of some inventory stressors or loads is not considered on potential endpoint or midpoint categories. The spread of invasive species due to anthropogenic activities serves as one example. The challenge is to determine the circumstances under which species invasion and spread is a relevant damage category for the biotic natural environment (Jolliet et al. 2004). Until recently, traffic noise provided another example of this problem. Jolliet observed that quantitative impact pathways to possible midpoints or human health damage have not been developed and LCI data for traffic noise is lacking (Jolliet et al. 2004). However, Mueller-Wenk developed a method connecting traffic noise and human health effects using data for accidents in Switzerland (Mueller-Wenk 2004).

2.1.5 Potential dangers of using predefined LCIA methodologies

On many occasions, a LCA practitioner may simply select a LCIA methodology provided as part of a LCA software tool. In these cases, the impact category, indicator and model selection, and classification have been preselected for the user. This is appealing from the practitioner’s point of view since it is faster and less costly. However, it must be noted and cautioned that depending on the methodology chosen and the impact categories of interest, the user may obtain qualitatively different results. For example, in a study comparing three LCIA methods, two midpoint-based approaches (EDIP97 and CML2001) and one endpoint-based approach (Eco-Indicator 99) it was concluded that for the midpoint approaches, if the chemical impacts on human health and ecosystems are important it matters which method should be chosen (Dreyer et al. 2003). For EDIP97 and Eco-Indicator 99 it was speculated that “...the two methods may produce diverging results, were they to be applied to comparisons of other types of products” (Dreyer

et al. 2003). Implicit weighting through uneven subcategorization might also lead to qualitative differences. For instance, TRACI uses different numbers of subcategories for different categories (Bare et al. 2002); equal weighting of each subcategory biases the result toward categories with more than one subcategory. Finally, some of these methods may not consider impact categories that are currently debated, which may be relevant to the LCA study being conducted.

2.2 Spatial variation

During LCA's codification in the 1990s, researchers came to appreciate the importance of spatial considerations (Bare et al. 1999; Graedel 1998; Hauschild and Wenzel 2000; Owens 1997b; Tolle 1996). Emissions generated by a product's life cycle occur at many locations, enter multiple media (air, water, land), and cause impacts in relation to local environmental sensitivities (Owens 1997b; Reap et al. 2003). Unlike global impacts such as stratospheric ozone depletion and global warming, those affecting local, regional and continental scales require spatial information in order to accurately associate sources with receiving environments of variable sensitivity. Yet, most assessments continue to ignore spatial considerations despite a decade of awareness and method development meant to correct these problems (ISO 2000a; Potting and Hauschild 2006). The failure of spatially explicit methods to penetrate LCA practice suggests the need for continued efforts and motivates this portion of the review.

When discussing the limitations placed upon LCA by ignoring space, one usually uses the following classification scheme to describe different types of assessments (Potting and Hauschild 2006):

- *Site-generic* LCAs lack spatial information and assume globally homogeneous effects.
- *Site-dependent* LCAs use varying spatial resolutions for emission and deposition sites.
- S*ite-specific* assessments model individual sources and local responses.

To better illuminate the connection between cause and effect, this review partitions the spatial problem differently. Instead of site-dependent and site-specific, this review discusses the problem in terms of spatial variation and local environmental uniqueness. Spatial variation refers to differences in geology, topography, land cover (both natural and anthropogenic), and meteorological conditions. Local environmental uniqueness refers to differences in the parameters describing a particular place (i.e., soil pH). This section discusses problems associated with spatial variation.

2.2.1 Spatial variation and transport media

Site-generic LCAs become less accurate when spatially variable transport phenomena begin to dictate the fate of health and environmental stressors. Spatially explicit modeling using versions of the RAINS model (150× 150 km grid squares) revealed that meteorological variations and local environmental sensitivities cause three order of magnitude acidification and eutrophication differences between European regions (Huijbreghts et al. 2001; Potting et al. 1998). Meteorological conditions and population distributions caused differences in health effects and damage to the 'man-made environment' for different European countries when airborne pollutant emissions were modeled using the EcoSense model (50×50 km grid squares) (Kerwitt et al. 2001). Later modeling efforts indicated that country-dependent acidification factors span a two order of magnitude range and that finer spatial resolution is needed (Hettelingh et al. 2005).

Atmospheric variations are not the only spatial variations capable of influencing life cycle impact assessments. Topography and hydrology also play a part. Region specific impact score formulations for airborne deposition of eutrophying compounds contain factors for runoff, a factor strongly influenced by topography (Huijbreghts and Seppala 2000). Estimates of water quantity impacts (i.e., drought stress on biomass, well failure, etc.) depend upon spatially explicit hydrology models or data sets (Heuvelmans et al. 2005; Reap et al. 2004). Groundwater contamination from landfills has been found to vary by as much as four orders of magnitude based on geological conditions and geographic location (Hellweg 2001).

2.2.2 Spatial variation and land use

Changes in land use and land forms directly and indirectly affect LCAs. Some land use impacts fall within existing impact categories while others may require new categories (Udo de Haes 2006). Infrastructure supporting a product's life cycle (mines, farms, factories, road networks, landfills, etc.) occupies ecologically productive land, potentially leading to reductions in biodiversity, loss of biotic production, and soil quality (Canals et al. 2006). Land use indirectly affects assessments by changing meteorological and hydrological patterns. Land cover alterations lead to regional climate changes by effecting net radiation and the division of precipitation (i.e., runoff from impermeable surfaces) (Foley et al. 2005). Precipitation division as well as diversion to agricultural, industrial, and domestic consumption influence the hydrological cycle (Foley et al. 2005).

"Land use impacts are highly dependent on the conditions where they occur..." (Canals et al. 2006), and the

spatial patterns of occupation influence ecosystem function (Turner et al. 2001). In Lindeijer's review of land use impact methodologies, the "functional approach" to impact assessment contains land functions such as 'erosion resistance' that partially vary with topography, for example (Lindeijer 2000). Even simple spatial variations in anthropogenic land forms, such as stack height, cause differences in estimated exposure efficiencies and health impacts (Finnveden and Nilsson 2005; Nigge 2001b).

The spatial nature of both direct and indirect influences highlights the importance of explicitly modeling land use patterns. Indeed, those developing characterization factors for land use impacts recommend biogeographical region scale differentiation, at a minimum (Brentrup et al. 2002; Canals et al. 2006).

2.3 Local environmental uniqueness

The problem of space in LCA encompasses more than variations in geological, topographic and meteorological geometry. Each environment affected by resource extraction or pollution is, to a greater or lesser extent, unique. As a result, each local environment is uniquely sensitive to the stresses placed upon it by a particular product system's life cycle.

Evidence of environmental uniqueness' importance comes in the form of investigations of the influence of site-specific data and research programs directed at including local sensitivities in LCA. Ross and Evans found that simply disaggregating inventory data to allow site-specific estimates for photochemical smog clarified impact assessment (Ross and Evans 2002). Part of the previously mentioned multiple order of magnitude differences between acidification, eutrophication, and health effect factors among European regions resulted from variations in local sensitivities (Huijbreghts et al. 2001; Kerwitt et al. 2001; Potting et al. 1998). In the case of acidification, some of the difference almost certainly stemmed from the fact that soil buffering capacity varied across the analyzed regions. Human health can also prove locally unique in a particular impacted environment. For example, parameterizing urban landscapes by population density influenced estimates of air pollutant exposure efficiencies and revealed that generic assessments can underestimate the impacts of particulate emissions (Nigge 2001a, b). Population density proved a significant factor in determining impacts caused by vehicle and power plant emissions in other studies as well (Finnveden and Nilsson 2005; Moriguchi and Terazono 2000).

Modeling the geometry of extraction and pollution as discussed in Section 2.2.1 is an important step, but fully accounting for the problem of space in LCA requires attention to the parameters that further define unique, impacted areas. Identification and inclusion of these parameters and associated models represent an area of continuing research in LCA.

2.4 Dynamics of the environment

"LCA is primarily a steady-state tool..." (Udo de Haes 2006) that typically excludes temporal information (ISO 2000a). Unfortunately, industrial and environmental dynamics affect impact assessment (Field et al. 2001; Graedel 1998; Owens 1997b). ISO 14042 acknowledges that ignoring time reduces the 'environmental relevance' of at least some results, but it does not discuss the inherent problems causing this reduction in relevance (ISO 2000a). This section highlights a number of the problems caused by ignoring system dynamics in life cycle assessment.

Temporal factors such as timing of emissions, rate of release, and time-dependent environmental processes affect the impact of pollution (Owens 1997b). For example, a volatile organic compound release timed to coincide with daylight hours produces more photooxidants than the same amount released at a constant rate during a 24-h period (Graedel 1998). Time-dependent environmental processes such as pollutant accumulation lead to threshold violations and accompanying variations in ecosystem impacts. Acidification impacts change when an ecosystem's nitrogen holding capacity is exceeded, for instance (Udo de Haes et al. 2002). Other time-dependent processes require years of chronic pollution before manifestation occurs; decades of critical loading can be required before terrestrial eutrophication impacts appear (Udo de Haes et al. 2002). Emissions generated by a life cycle under examination might synergistically interact with background pollution, leading to worse impacts that expected. Impacts such as aquatic eutrophication demonstrate seasonal variation in particular regions (Udo de Haes et al. 2002).

In some cases, temporal patterns in product production, use, and disposal also might influence the accuracy of life cycle assessments. Taking a fleet-centered approach to life cycle assessment, Field and coauthors theoretically demonstrated that transient environmental impact differences could dominate steady-state differences if one selected a short enough time horizon for evaluation (Field et al. 2001).

Lacking dynamic representations or historical data, traditional life cycle assessment cannot account for environmental and industrial dynamics. Changes in pollution profiles as well as ecosystem responses are averaged, and impacts with sufficiently long delays may even be ignored. Responses to environmental interventions cannot be accurately modeled. Potentially important industrial transients receive no attention. Amelioration of these problems will undoubtedly improve LCA's environmental relevance.

2.5 Time horizons

Time presents a problem for life cycle assessment apart from industrial and environmental system dynamics. LCA integrates environmental impacts over time. To obtain this integrated value, one must select an appropriate time horizon. Discussions about appropriate time horizons are not merely academic. A life cycle assessment of different incineration technologies found that the technology with the most favorable environmental profile changed with temporal system boundary (Hellweg 2001). Selecting integral limits and valuing impacts distributed in time are problems of continuing discussion in the LCA community.

With regard to integration, this discussion centers on the appropriateness of infinite vs. finite limits (Udo de Haes et al. 1999). Infinite limits capture the entire effect of an impact, but such limits effectively discount short-term impacts (Udo de Haes et al. 2002). Furthermore, realistic integration over infinite time may prove challenging. While avoiding inaccuracies associated with extrapolating into the distant future, a finite limit effectively discounts long-term impacts by truncating the period of consideration. One also faces the problem of selecting an appropriate finite limit, which might differ by impact category (Udo de Haes et al. 1999). This limit could be chosen arbitrarily, or as Canals and coauthors suggest for land use, one might select the time required for a perturbed system to achieve a steady-state (Canals et al. 2006). Of course, the latter limit assumes the existence of some method for determining the time to steady-state. Valuation within the context of time horizon selection takes two forms. Implicit valuation occurs when one chooses to truncate impact consideration or allow infinite limits as previously mentioned. Explicit valuation occurs when one weights the value of an impact by time. In other words, explicit valuation occurs when one applies a discount rate to future impacts as one would discount future cash flows in a financial analysis (Udo de Haes et al. 1999). Choosing an appropriate environmental impact discount or appreciation (the 'discount' rate could be negative) rate is not a matter free of controversy.

3 Problems in interpretation

The complexity of interpretation depends on whether the assessment objective, defined at the outset of LCA, is to target improvements ('what is bad'), to recommend a course of action (e.g., a strategy, prioritization of efforts, or choice of design), or to determine objectives for a more in-depth LCA. This section discusses interpretation from the perspective of decision making more so than quantitative analysis. Aggregation is unavoidable when recommending one among a set of possible actions, and dealing with aggregation means dealing with the pernicious problems of weighting and valuation. The need to impute socially rooted values in the form of weights places this difficulty in the same class as problematic decisions discussed during the review of goal and scope problems in Part 1. Even if an assessment does not involve selection, one must still account for and manage uncertainty in the decision process. This section discusses problems in the interpretation phase stemming from weighting, valuation and uncertainty management.

3.1 Incorporating subjective values using weighting

Decision makers often consider multiple objectives that conflict or *trade-off* across a set of decision options: one option dominates the others for one objective but is itself dominated for another objective. To identify the most preferable decision option, one relatively weights the importance or value of different objectives and aggregates them into a single composite score. For LCA, this requires quantifying and comparing the value of different environmental impacts even when their units and scales differ. To address this fundamental challenge, numerous weighting methods having different preference elicitation processes have been proposed and applied. The type and amount of effort required by an elicitation process is proportional to the number of objectives, requirements for accuracy and rationality in preference structuring, and number of stakeholders involved. Finnveden and coauthors conducted a comprehensive survey of these methods and evaluated them using a variety of performance criteria (Finnveden et al. 2002). No weighting method met all of the authors' criteria; in fact, they theorized that none ever would (Finnveden et al. 2002). The use of weights can pose a challenge for two general reasons:

1. It may be difficult to assure that an elicited weight accurately reflects a decision maker's value for some performance objective, particularly with respect to other performance objectives.
2. Weights derived through different value (or preference) elicitation methods may not be comparable, and therefore, aggregation may not be possible without hiding added assumptions.

Particular instances of these problems are next discussed for the two broadest categories of valuation: monetization and all other methods.

3.1.1 Problems in monetization methods

Coming from environmental economics, monetization methods use a monetary measure to quantify value for all environmental impacts (Turner et al. 1993). Numerous authors have discussed theoretical and empirical concerns with assigning economic value to environmental aspects

(Bockstael et al. 2000; Farrow et al. 2000; Kahneman and Knetsch 1992). Only the most prominent of these problems are presented in this paper. Most monetization methods measure willingness to pay (WTP). However, *who* is paying, *how* that is measured, and *what* values are actually involved may vary. While some methods ask individuals directly, others assume WTP is revealed in market behavior; likewise, WTP may be based on the environment's use value, nonuse value, or total economic value (Finnveden et al. 2002; Turner et al. 1993). Although the results of these different monetized measures share the same units, some authors warn that they should not be considered comparable or additive, nor should they be combined with nonenvironmental costs (Bockstael et al. 2000; Finnveden et al. 2002).

There is also a controversy about whether monetization methods alone are comprehensive enough. Some authors advocate less reliance on WTP in general, noting that they fail to account for regional differences and externalized costs (Matthews and Lave 2000). Though some LCA researchers are strong proponents of using market value, arguing its primacy over energy or other theories of value (Ayres 1995), others contend that other forms of valuation provide a balance and must be considered (Ehrenfeld 1997). For human health valuation, WTP relies on significantly different values than other measures such as quality-adjusted life years and therefore could lead to different decisions (Hammitt 2002).

Determining a timescale for valuation is another major problem affecting monetization methods. To monetize future impacts, one must choose between discounting them (either through choice of a rate, or implicitly in willingness-to-pay methods) or simply ceasing to value them past a certain time (Finnveden et al. 2002). Hertwich, Hellweg and coauthors discuss discount rate problems in greater detail (Hertwich et al. 2000; Hellweg et al. 2003). When monetized impacts are aggregated, choices of discounting may be mixed and not transparent.

3.1.2 Problems in other weighting methods

The field of decision analysis produced a variety of methods that achieve weighting. Though techniques vary, they typically involve normalization[1] to give different objectives (or metrics) compatible units, followed by weighting and aggregation of those normalized scores. Although theoretically promising, weighting methods rooted in decision analysis still suffer from problems. First, when preferences are constructed by direct elicitation, the resulting weights or value functions may be affected by bias, some of which is unavoidable. Root causes of bias may be of a behavioral type (Weber and Borcherding 1993) or a procedural type. Procedural bias can be that due to survey framing or wording (Mettier and Hofstetter 2004) or to choice of cognitive references (Mettier et al. 2006). Similarly, preferences elicited through indirect methods (i.e., revealed) may be of questionable relevance when transferred to the actual context of interest (Finnveden et al. 2002). Because the existence of bias is difficult to verify, it is difficult to measure whether it has been reduced to acceptably low levels (Finnveden et al. 2002). Bias also affects normalization, which can be used as a precursor to weighting (Heijungs et al. 2007).

The need to assign weights between objectives poses additional challenges. Methods based on multi-attribute value (or utility) theory provide a rational axiomatic foundation for eliciting preferences for trade-offs (Keeney and Raiffa 1976). However, their elicitation processes require a large number of steps (Hertwich and Hammitt 2001). Assigning simple weights, on the other hand, may be less accurate, does not ensure rationality in applying preferences, and may suffer from anchoring biases (Mettier and Hofstetter 2004). Similarly, distance-to-target weighting methods do not prescribe how to set targets, nor do they enforce equal importance for each objective, a requirement for intereffect weighting (Finnveden et al. 2002). In general, for all the aspects of valuation previously mentioned, whether an approach or its resulting preference model is 'good' or not depends on the importance and complexity of the scenario, which in some cases are difficult to gauge. This fact impedes the creation of a universal framework for choosing weighting methods with the most appropriate level of complexity.

The preceding discussion on weighting focused on supporting selection between alternatives; however, ideally, weights could be used to assess whether, on a standalone basis, a product system is environmentally 'good,' 'bad', or even 'sustainable.' Quantifying an absolute, aggregate measure in these terms is complicated by most of the weighting problems presented previously. Moreover, it depends on better definition of concepts of acceptable environmental performance and sustainability.

3.2 Uncertainty in the decision process

Whether the desired outcome of a LCA is a simple benchmark or a more involved recommendation of action,

[1] This section discusses a type of normalization which is an 'operational prerequisite to weighting', also labeled 'case-specific normalisation' by Finnveden et al. (2002). However, a different definition of normalization, not discussed in this section, is given by ISO 14044: comparison of the magnitude of indicator results to information from some *external reference system*, e.g., some sector, temporal span, or spatial region (ISO 2006b). Defining such a reference system is a methodological decision point, one where sensitivity analysis might be warranted. A potential problem is the lack of consensus on standards or guidelines for defining a reference system, even for a specific industry or sector.

its reliability depends on appropriate consideration of uncertainty. Uncertainty is lack of knowledge about the true value of a quantity, true form of a model, appropriateness of a modeling, or methodological decision, etc. One evaluates the effects of uncertainty using two main classes of techniques that will be referred to repeatedly in this section. The first, *uncertainty analysis*, models uncertainties in the inputs to a LCA and propagates them to results. For comparative LCAs, this can reveal whether there are significant differences between decision alternatives. The second type, *sensitivity analysis*, studies the effects of arbitrary changes in inputs on LCA outputs. This helps to identify the most influential LCA inputs when their uncertainty has yet to be or cannot be quantified.

The ISO LCA series of standards briefly mentions these two techniques but provides little guidance (ISO 2000a, b, 2006b) as to when or how procedurally to apply them. In response, LCA researchers and practitioners have proposed or adopted different variations of these techniques (Björklund 2002; Lloyd and Ries 2007). Choosing one can be difficult, especially for predictive assessments, comparative assessments of complex systems, or assessments with broad scope. Even when an appropriate choice is apparent, in practice, there are still many hurdles to using them.

In this section, problems associated with evaluating uncertainty in LCA fall into four categories:

- Modeling of uncertainty
- Incorporation of multiple uncertainties
- Completeness of analysis
- Cost of analysis

A review of the root causes of data quality limitations is presented later in Section 4.

3.2.1 Appropriate representation of a given uncertainty

Mathematically representing the variety of uncertainty types identifiable in LCA is often not straightforward. Probability distributions can be used to represent random variability in input parameters, arguably the most familiar type of uncertainty in LCA. Probability distributions have been used in a variety of LCA uncertainty analysis methods (Björklund 2002; Ciroth et al. 2004; Huijbregts 1998) and applied to numerous case studies (Geisler et al. 2004; Maurice et al. 2000; McCleese and LaPuma 2002). However, Björklund observes that few classical statistical analyses in the LCA literature describe their data sources or assumptions or reveal how probability distributions were determined (Björklund 2002). Probability distributions can, alternatively, be defined based on subjective expert estimates rather than data sets; in fact, this option is used for a majority of LCI data (Björklund 2002). In such cases, estimates and assumptions about probability distributions should be subjected to sensitivity analysis to ensure credibility, thereby adding a layer of complexity to the analysis.

Apart from variability, another chief source of uncertainty in information or models relates to their *representativeness*, commonly limited due to missing or incomplete data (Weidema and Wesnæs 1996). In response, a variety of mathematical formalisms, also surveyed by Björklund (2002), have been proposed for use in uncertainty analysis, including, possibility distributions (Benetto et al. 2005), upper and lower bounds with no distributional information (Pohl et al. 1996), fuzzy intervals (Gonzalez et al. 2002; Güereca et al. 2007; Pohl et al. 1996; Sadiq and Khan 2006), and information gaps (Ben-Haim 2006; Duncan et al. 2007). None are clearly most appropriate for every uncertain phenomenon and every LCA scenario. Also, some types of uncertainty, such as that due to LCA methodological choices, may not be representable using any uncertainty formalism and may need to be evaluated instead using sensitivity analysis (Björklund 2002; Lloyd and Ries 2007). Overall, we expect it could be difficult for novice practitioners to select from the variety of uncertainty formalisms. This is a problem because the results of an uncertainty analysis can vary depending on one's choice of uncertainty formalism.

3.2.2 Incorporation of multiple heterogeneous uncertainties into final LCA results

Problems also become apparent when one attempts to aggregate, for decision purposes, the influence that multiple heterogeneous uncertainty types have on LCA results. This is particularly problematic for comparative LCA, where the goal is identification of the best performing alternative, even for a single environmental performance dimension. In best case scenarios where all input uncertainty can be represented by probability distributions, uncertainty can be propagated to outputs using well-established techniques. From there, a decision maker can compare statistical differences or expected (i.e., average) environmental performance.

However, in one LCA alone, it is possible that one or more uncertainty representations other than probability distributions are warranted due to the sparsity or non-probabilistic nature of available information. Unfortunately, combination of different uncertainty formalisms is often mathematically impossible and, when feasible, not theoretically sound, though this capability is being pursued by some researchers (Joslyn and Booker 2004). This prevents the incorporation of all uncertainty types into a single propagated result, even for one environmental performance dimension. Given the convenience of such single 'scores,' practitioners might be tempted to model all uncertainty information using a single formalism unjustifiably, either

relying on unwarranted assumptions or ignoring available data. Even though ISO mandates that assumptions be documented (ISO 2006b), detecting whether or not such assumptions lead to an unreliable decision could be difficult for a complex assessment case.

In some cases, qualitative information may be available to describe the degree of representativeness of uncertain quantities or models. Examples of this metadata (i.e., data about data) include dimensions such as age of the data, the geographical area to which it applies, and technology assumptions. Researchers have proposed formalizing these metadata types as data quality indicators (DQI) (Weidema and Wesnæs 1996), though opinions differ as to how to incorporate them. ISO 14041 (ISO 1998) only recommends providing such metadata alongside LCA results for transparency purposes or to guide which alternative scenarios to analyze using sensitivity analysis as defined by ISO. Weidema and Wesnæs have proposed a method for transforming DQI scores to probability distributions using predefined, default distributions (Weidema and Wesnæs 1996); however, these conversions are subjectively defined.

To summarize, the fundamental problem is a trade-off between two aspirations. The first is the (idealistic) motivation to utilize as much available information (qualitative or quantitative) about uncertainty—and as few unwarranted assumptions about that information—as possible. The conflicting aspiration is to factor all uncertainty models, however heterogeneous in form, into an efficient, rational decision-making process. To date, there are no frameworks for uncertainty analysis in LCA that guide characterization of this trade-off to make the assessment as comprehensive as possible yet still tractable in terms of decision making.

3.2.3 Completeness and conclusiveness of LCA uncertainty analysis

Intuitively, limitations in the comprehensiveness of an uncertainty analysis can considerably affect the quality of LCA conclusions and recommendations. The level of completeness achievable is proportional to the scope defined for a particular LCA, e.g., the time and geographical boundaries chosen. For complex products with long lifetimes, a 'complete' characterization of uncertainty might only be possible over a timescale that is too small to be of use to the practicing organization.

In addition, the degree to which comprehensiveness can be achieved (e.g., direct data collection, quantification of uncertainty in representativeness, model validation, etc.) varies across the phases of a LCA. For instance, developing models and characterizing uncertainty tends to be harder for impact assessment than for life cycle inventory (Owens 1997b) and, likewise, harder for some indicator categories than others (ISO 2000a, 2006b).

Even if one can achieve comprehensiveness in some portions of a LCA uncertainty analysis, the severe uncertainty and data limitations of other more difficult portions can dominate LCA outcomes and lead to inconclusive outcomes (Björklund 2002; ISO 2000b, 2006b). In response, practitioners can be tempted to characterize more readily quantifiable uncertainty and fail to acknowledge (or even know about) the existence of other uncertainty (Finnveden 2000). Such partial uncertainty analyses may generate false confidence in the reliability of results (Bare et al. 1999).

For comparative LCA, a converse problem also arises: modeling uncertainty in all LCA phases comprehensively and conservatively can lead to inconclusiveness. A complete representation of uncertainty may entail wide probability distributions or broad intervals of imprecision, propagating to results to make the alternatives under consideration indistinguishable. In fact, Finnveden argues that, from a scientific perspective, "it can in general not be shown that one product is environmentally preferable to another one, even if this happens to be the case" (Finnveden 2000).

The above problems of LCA have also been considered in the field of risk assessment. Risk assessments (or analyses) in general tend to rely on specific models of the mechanisms related to risks, usually valid for a specific place and time (Dekay et al. 2002; Morgan et al. 1990). In contrast, LCA tends to include multiple impacts over different temporal or spatial scales, often with simplified models or assumptions. From the RA perspective, the lack of "spatial, temporal, dose-response, and threshold information" in LCA makes its results overly conservative, since it implies that all environmental burdens will affect sites that are sensitive to adverse impact (Owens 1997a). Direct comparisons between RA and LCA have clarified where the techniques are compatible or overlap or where their respective practitioners could learn from each other (Cowell et al. 2002; Hofstetter et al. 2002; Matthews et al. 2002). However, limited conclusiveness in results remains a problem in LCA due to its often wide scope.

3.2.4 Resource intensiveness of LCA uncertainty analysis

Lastly, the analysis of data quality in LCA, including sensitivity analysis and uncertainty analysis, incurs costs that can be daunting to practitioners. Deriving probability distributions through statistical analysis requires significant collection of test data. Alternatively, subjective distributions can be defined based on expertise, but better data requires more knowledgeable and expensive experts. The costs of characterizing uncertainty (by whatever means) is generally

not quantified nor discussed in LCA research or practitioner communities, nor are techniques or frameworks that guide efficient data gathering. Admittedly, such issues would be hard to generalize for all LCA types.

While computation in uncertainty analyses (e.g., for Monte Carlo simulations) is consistently becoming cheaper and faster, the problem scenarios that require this computation can still be complicated to formulate. For instance, sensitivity analysis, whether used for uncertain methodological choices or for evaluating the effects of unrepresentativeness, can be effort intensive if a large number of scenario combinations must be defined, separately evaluated and compared (Björklund 2002).

4 Data quality—a problem affecting all LCA phases

Having reviewed the challenges associated with making decisions under uncertainty, attention is next given to the main reasons for that uncertainty: data or models that are of poor quality. In her survey of approaches to improve reliability, Björklund generally identifies the main types of uncertainty due to data quality: badly measured data ('data inaccuracy'), data gaps, unrepresentative (proxy) data, model uncertainty, and uncertainty about LCA methodological choices (Björklund 2002). Specific instances of these data quality limitations are next discussed, grouped by those that are general, those that specifically affect life cycle inventory and those particular to impact assessment.

4.1 General problems limiting data quality

A number of general reasons explain the existence of poor or unavailable data. Data and models alike can fail to accurately represent the full spatial and temporal scope chosen in the initial phase of a LCA. Data can be effectively unobservable during the time period devoted to conducting a LCA. For example, consider product recovery infrastructure models and scenarios. Similarly, a LCA practitioner may not even recognize the need to collect some data. Uncertainty may also arise when different data sources measuring the same quantity conflict (Björklund 2002; Finnveden 2000). Standardized databases of LCA data are sought to reduce the burdens of data collection; yet, easily accessible, peer-reviewed data sets remain absent (UNEP 2003). There are few established, standardized, or consistent ways to assess and maintain data quality (Vigon and Jensen 1995). Regarding LCA databases, Bare and coauthors identify a fundamental conflict between the sophistication of the data and the variety of categories that the data covers (Bare et al. 1999).

4.2 Data quality in LCI

Some barriers to data collection are specific to inventory analysis. In general, the literature tends to agree that data for life cycle inventories is not widely available nor of high quality (Ayres 1995; Ehrenfeld 1997; Owens 1997b). Data collection costs can be prohibitively large, e.g., when submetering must be implemented in an industrial facility, when data must be gathered from the field or when data must be frequently collected to remain relevant (Maurice et al. 2000). In other cases, data exists outside of the LCA practitioner's organization, e.g., when withheld upstream or downstream by suppliers or other partners who have concerns (potentially valid) that sharing inventory data might reveal confidential information related to their competitive advantage (Ayres 1995). When available, external data can be of unknown quality. When data is not measured by the organization conducting the LCA, the accuracy, reliability, collection method, and frequency of measurement may not be known and the limits of the data cannot necessarily be deduced (Lee et al. 1995). As a result, uncertainty distributions or even upper and lower bounds are commonly unavailable (Owens 1997b). Furthermore, mass balances are often not performed, or are performed incorrectly (Ayres 1995). Data also can become outdated, compiled at different times corresponding to different materials produced over broadly different time periods (Jensen et al. 1997). LCI data may be unrepresentative because it is taken from similar but not identical processes, is based on assumptions about technology levels, or uses averages, all of which may be features of database values (Björklund 2002). During inventory analysis data with gaps are sometimes ignored, assumed or estimated (Graedel 1998; Lent 2003). Also, practitioners may extrapolate data based on limited data sets (Owens 1997b). In fairness, it should be noted that ISO LCA standards require a company to document its data sources (ISO 2006a, b), addressing many of the concerns raised in publications written in the late 1990s. Still, companies not complying with ISO might take these shortcuts, limiting data quality.

4.3 Data quality in impact assessment

Probably the most serious data and model quality limitations affect the impact assessment stage, as there tend to be large discrepancies between a characterization model and the corresponding environmental mechanism (ISO 2000a, 2006b). The most fundamental barrier to model quality are limits to available scientific knowledge (ISO 2000a, 2006b). Invasive natural species and genetically modified species change ecosystem compositions and functions in ways that may prove difficult to foresee (Jolliet et al. 2004). New chemicals constantly appear on the industrial market with

poor models or measures of the mechanisms that disperse them into the environment. Finnveden points towards this being the case with dioxins (Finnveden 2000). Even if dispersion models exist, fate still may be ignored in calculations of impact (Bare et al. 1999). Besides dispersion, the threshold levels that would create environmental damages may not be modeled or measured (Owens 1997b) or may be represented using reduced order models, such as with linear dose-response curves (Bare et al. 1999). Even if thresholds are known, they might not apply to any particular locale or time period, or they might be affected by synergistic combinations of chemicals (Bare et al. 1999; Björklund 2002). To summarize the fundamental problem of modeling to an appropriate level of comprehensiveness, especially for environmental impact assessment, Bare and coauthors note that it is hard to know "where to draw the line between sound science and modeling assumptions" (Bare et al. 1999).

5 Critical problems and recommendations for future research

Having reviewed many of the problems suffered by LCA, we offer our opinion of their severity, the adequacy of current remedies, and areas requiring particular research attention. We also propose research agendas meant to address the identified critical problems.

5.1 Ranking problems in LCA

Table 1 summarizes our opinions about severity and solution adequacy using a simple ordinal scale. Each number represents a qualitative estimate. Severity represents a combination of problem magnitude, likelihood of occurrence, and chances of detecting the error should it occur. For instance, spatial variations can lead to multiple order of magnitude differences in characterization factors for commonly used impact categories such as acidification. Given the distributed nature of these impacts and the effects of other contributing sources of pollution, it would be difficult to detect the failings of a homogeneous approach without the aid of spatial models. Therefore, we rate the spatial variation problem a 5.

Table 1 Problems in LCA qualitatively rated by severity and adequacy of current solutions (1, minimal severity while 5, severe; 1, problem solved while 5, problem largely unaddressed)

Problem	Severity	Solution Adequacy
Functional unit definition	4	3
Boundary selection	4	3
Social and economic impacts	3	4
Alternative scenario considerations	1	5
Allocation	5	3
Negligible contribution criteria	3	3
Local technical uniqueness	2	2
Impact category selection	3	3
Spatial variation	5	3
Local environmental uniqueness	5	3
Dynamics of the environment	3	4
Time horizons	2	3
Weighting and valuation	4	2
Uncertainty in the decision process	3	3
Data availability and quality	5	3

Solution adequacy integrates capacity to address the discussed problem and difficulty of using available solutions. For example, ISO's suggested approach to dealing with boundary selection involves truncation errors and significant subjectivity while IO LCA techniques introduce significant additional complexity (see Part 1). Available guidelines and tools represent significant progress, but the boundary problem remains unresolved. So, we rate its solution adequacy a 3. We believe that functional unit definition, boundary selection, allocation, spatial variation, local environmental uniqueness, and data availability/quality are critical problems requiring particular attention.

5.2 Archetypes and templates for goal and scope and LCI

Critical problems appear in each LCA phase. Functional unit definition and boundary selection seem the most severe in the goal and scope phase. We take this position because failure to select a representative functional unit or properly define a study's boundaries can reduce a LCA to a misdirected and expensive exercise in data acquisition. Allocation seems the most pressing problem in inventory analysis. We observe that allocation failures hide or exaggerate burdens associated with a product system, effectively biasing all downstream results with an artifact of the analysis. The LCA community appreciates these problems, and it has devoted no small effort to addressing them. As discussed in Part 1, however, a consensus on the most appropriate remedy remains elusive.

Given that multiple rational approaches to these problems exist and contribute to the inability to form a consensus around one best approach, we suggest development of LCA archetypes for functional unit definition, boundary selection, and allocation. We propose a research program of an integrative nature. It would classify types of LCAs and associate the most appropriate approach to the mentioned problems with each class.

5.3 Spatially explicit, multiscale, multimedia models for LCIA

Impact assessment accuracy suffers most when one ignores spatial variation and local environmental uniqueness.

Multiple order of magnitude differences in characterization factors observed for different regions support this contention (see Sections 2.2 and 2.3). Given such large differences, the credibility of assessments using locally and regionally significant impact categories hinges upon accurate representation of regional uniqueness. Therefore, development of spatially explicit models capable of accounting for local environmental uniqueness should be a research priority. These models should consider spatial scales smaller than regions and allow improved resolution around major pollution sources, deposition areas and sensitive zones (i.e. population centers, productive ecosystems and protected areas). Even when utilizing available models and data, multiscale, multimedia modeling of environmental stressors is an onerous task requiring a sustained commitment often only supported by governmental and international organizations. We also recognize that exercising such complex models may prove too difficult or costly for those working in product design. Therefore, we suggest using models generated by national and international groups as calibration standards for less sophisticated, more tractable tools for industry.

5.4 Standardized, peer-reviewed databases and model bases

Finally, a serious effort must be undertaken to improve the availability and quality of LCA data. We strongly support calls to create standardized, peer-reviewed LCI databases and 'test-bed' data sets (Thomas and Graedel 2003; Thomas et al. 2003). National Institute of Standards and Technology's publicly available 2D and 3D CAD model repository might serve as an instructive example of one such standardized database for test data (Regli and Gaines 1997). These data should include uncertainty distributions, descriptions of collection methods, sampling frequency, and dates at a minimum. Provisions for updating these databases with information for new materials and processes should be devised and implemented. Additionally, input–output tables correlating economic and environmental flows for countries increasingly involved in global trade should be generated and maintained. These recommendations also apply to impact assessment, though implementing them may prove more difficult and costly.

Taking the position of an engineering designer, we believe an unmet need for LCA model bases exists. Model bases store basic physical and empirical relations for activities in product systems. In place of data points, they hold models capable of producing ranges for LCI data. For example, in place of energy consumption and environmental burden data for a particular class of truck, one would find a model relating truck velocity and total mass to energy consumption and emissions. Such models allow tailoring of life cycle inventories to analyzed product systems. They might also allow designers to incorporate more parameters into design problems, increasing their capacity to explore a product system's design space.

6 Summary

LCA offers a rational and comprehensive approach to environmental assessment of product systems. However, it is clearly not flawless. Each phase suffers from problems that degrade accuracy and increase uncertainty of assessment results. It shares some problems with other analysis techniques (i.e., boundary selection), others are classic problems in LCA (i.e., allocation) and the importance of still others emerged in the last decade (i.e., spatial variation/site-dependency). While methods meant to mitigate these problems exist, a final judgment on the most appropriate frameworks remains elusive. Considering the ultimate purpose of LCA, resolution of many, if not all, of these problems is of paramount importance. Problems with functional unit definition, boundary selection, allocation, spatial variation, local environmental uniqueness, and data availability/quality deserve particular attention.

Problems encountered during goal and scope definition arise from decisions about inclusion and exclusion. When defining functional units, one must select the functions to include and settle upon ways to quantify them. Boundary selection requires decisions about including and excluding processes. Exclusion of economic and social impacts in LCA sets fundamental limits on the comprehensiveness of the tool. And, choice of alternative scenarios influences decisions in the interpretation phase. These choices and processes used to make them currently reduce assessment accuracy and introduce uncertainty.

Inventory analysis problems involve flows and transformations. The allocation problem results from the need to accurately associate flows from a multifunctional process to each of its functions or the products being assessed. The criteria used to identify and eliminate ('cutoff') unimportant resource and waste flows become problematic when one attempts to balance information costs against the potential of missing substantial environmental effects. Local technical uniqueness becomes problematic when average or generic data or models are used to represent processes that significantly differ from the norm.

Truncations and assumptions about global homogeneity and steady-state conditions introduce the most severe errors in impact assessment. Selecting environmental impact categories effectively truncates the types of damages a study considers, thereby introducing inaccuracies. Setting arbitrary time horizons skews results in favor of short- or long-term impacts. Ignoring spatial variation, local uniqueness and environmental dynamics discounts the influence

of environmental stress concentrations, leading to inaccurate estimates of potential environmental damage.

Aggregation is the overarching problem in interpretation. Collapsing inventory or impact data into a single figure of merit requires weighting or valuation of some kind. Unfortunately, weighting and valuation introduce subjectivity, subjectivity that is not always satisfactorily handled by current methods in decision science. Inaccuracies in other phases and variability inherent in modeled systems result in a high degree of aggregated uncertainty by the time one reaches an LCA's interpretation phase. Making meaningful decisions under this potentially severe level of uncertainty is challenging.

In its current state, LCA provides relative and directional information to those working in environmental assessment, policy, design, and allied fields. If one accepts sustainability as the ultimate goal, the importance of improving LCA to the point where it offers more than ambiguous directional information is clear. Guided by LCA's current outputs, it is unlikely that one could avoid local minima of environmental damage that fall short of sustainability. Solving the discussed problems would improve LCA in this regard. A superior description of the complex environmental performance space for product systems might give those involved in numerous and varied decision processes the capacity to navigate society toward a more sustainable state.

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